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Testing of a Lagrangian model of dispersion in the surface layer with cattle methane emissions

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ABSTRACT

Results from an experiment measuring methane emissions from a herd of cattle are used to investigate the performance of a backward-Lagrangian stochastic model (distributed under the name WindTrax). The availability of simultaneous mass-budget measurements of the emission rate, together with a unique setup geometry, allow to compare modelled and measured normalised concentration profiles and horizontal flux profiles with five sensor heights, *z*, and for four horizontal source–sensor distances, *x*. Simulated emission rates differ typically by 10–20% to those obtained from the mass-budget measurements, which is in agreement with previous tests of the accuracy of WindTrax. Thus, the idealisation of a herd of animals as a homogeneous area source at ground level does not seriously affect the model's applicability to infer emission rates. The profile comparison suggests that WindTrax may overestimate the speed of vertical dispersion. As a consequence, for this experiment an ideal z/x ratio exists where the modelled emission rate is unbiased. Its value is about 0.080 in unstable and 0.067 in stable stratification. Using concentration measurements taken above or below this z/x threshold leads to emission rates that are slightly under- or overestimated, respectively. Simultaneous measurements with an open-path methane laser are compatible with this finding. Possible causes of the apparent overestimate of vertical dispersion rates are discussed, leading to the cautious suggestion that it may stem from the choices for the Kolmogorov constant and/or the normalised dissipation rate in the model, which reflects gaps in our understanding of the atmospheric surface layer. It is argued that this notion does not contradict the earlier results from a number of controlled tracer-release experiments that had been designed to test WindTrax.

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1. Introduction

"Lagrangian stochastic" (LS) models are the "natural and most powerful means" (Wilson and Sawford, 1996) to describe atmospheric dispersion processes. Such models can be run backwards in time in order to efficiently infer emission rates of gases from confined sources, using measurements of concentrations downwind of the sources as inputs. One particular implementation of such a backward-Lagrangian stochastic model (BLS) for the atmospheric surface layer is available under the name WindTrax (Thunder Beach Scientific, Nanaimo, Canada). WindTrax has recently become popular to estimate gas emissions from farm operations (see references later in this section). There is, therefore, a need to assess the performance of this model not only in ideal test setups, such as controlled-release experiments, but also in real-world farm situations. This study attempts such an assessment.

The mechanics of WindTrax have been described in detail by Flesch et al. (1995, 2004). At the core is Thomson's (1987) well-mixed three-dimensional LS model for Gaussian inhomogeneous turbulence. Wilson and Sawford (1996) point out that for the *vertical* dispersion in this type of turbulence, the Thomson model is the *unique* LS model. It contains one empirical parameter, which can be expressed either as a Lagrangian timescale or the Kolmogorov constant, as discussed by Wilson et al. (2001, 2009). In the horizontal dimensions, the Thomson model is not unique, and it is well-known that any model that uses only surface-layer parameters is prone to error because streamwise and lateral wind fluctuations contain larger-scale components originating outside the surface layer. For this reason, Flesch et al. (2004) considered line-averaged concentration measurements preferable to one-point measurements: provided that the line average includes the entire width of the gas plume, the BLS results will be insensitive to modelling error of horizontal dispersion.

Flesch et al. (1995, 2004) showed that WindTrax derives emission rates from measured concentrations with about 20% accuracy for a wide range of conditions in the atmospheric surface layer. This makes it arguably the best available model for this purpose

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in obstacle-free flow over flat terrain. Subsequently, WindTrax has been used to determine the trace gas emissions from livestock: of methane (CH_4) by Laubach and Kelliher (2005a,b), McGinn et al. (2006), Laubach et al. (2008), of ammonia (NH_3) by Denmead et al. (2004), Sommer et al. (2005), Flesch et al. (2007), McGinn et al. (2007), Harper et al. (2009), and of both gases simultaneously by Loh et al. (2008) and van Haarlem et al. (2008).

Laubach et al. (2008), henceforth L08, compared the BLS results to those of two other micrometeorological methods, using the same measured concentrations, wind speeds and turbulence parameters, and to an animal-scale tracer ratio method which determined the daily CH_4 emissions from each animal. The two other micrometeorological methods were an integrated horizontal flux (IHF) method (also known as mass-budget) and a flux-gradient (FG) method (with footprint correction accounting for the limited source area extent), both described in detail by Laubach and Kelliher (2004). Considering the animal-scale method as an absolute reference, L08 concluded that FG and IHF "provide unbiased means if the minimum distance between the animals and the instruments is of order 20 m", while BLS was "the most consistent method over time, but a bias of order 20% needs to be corrected for". This latter result is in qualitative agreement with two other experiments on CH_4 emissions from grazing cattle (Laubach and Kelliher, 2005a,b). However, it is at odds with the results of tracer-release experiments which showed no significant bias of the mean gas recovery rate (Flesch et al., 2004; Gao et al., 2009a). A somewhat more complicated picture emerges from the results of two other release experiments. First, McBain and Desjardins (2005) observed a dependence of gas recovery rates on measurement height, with emissions overestimated when using concentrations at the lowest height, and underestimated when using the uppermost height, respectively. In contrast to this, Laubach and Kelliher (2005a) had previously found the calculated emission rates to increase with the height of the concentration measurement, in three experiments with cattle herds. Second, Gao et al. (2009b) reported a stability dependence of the gas recovery rate, underestimating the true emission rate in unstable and overestimating it in stable stratification.

The present paper aims to understand whether such discrepancies can be attributed to the performance of the BLS model itself, i.e. the Thomson (1987) model, or whether they are due to operational differences between the tracer-release experiments and the animal-herd experiments. Possible factors are: differences in source geometries, differences in sensor geometries, and differences in the implementation of WindTrax. These are briefly described in the following.

First, there are some principal differences in the nature and distribution of the sources between the two groups of experiments. In the tracer-release experiments a true area source at ground level is created as well as possible, by a pipe grid releasing the trace gas uniformly in the horizontal dimensions, and constant in time. In the animal-herd experiments, the assumption of an area source is an approximation for a large number of point sources, the cattle, which can move freely within the confines of the paddock. Observations of the cattle behaviour show that they are generally evenly spread, but nevertheless the source distribution is patchier and less homogeneous in space and time than in the tracer-release experiment. In addition to the variability of emission *locations* due to animal movement, there is generic variability of emission *rates* between animals, and there are temporal variations related to the digestion processes. Another idealisation of the animal-herd experiments is that the emissions are assumed to be at ground level. While this is true for a major fraction of the emissions (whenever an animal exhales while its mouth is near the ground, either because the animal grazes or because it is lying down to sleep or ruminate), it is not true for all of the emissions, since the mouth of a standing cattle would typically be about 1 m above ground. A third difference between the two groups of experiments is the size of the source area, typically a few meter in each horizontal dimension for the tracer releases, compared with tens to hundreds of meters for the animal-herd experiments.

Second, the quoted tracer-release experiments and the animal-herd experiments differ principally in the distribution of concentration measurements. The latter employed *vertical profiles of point measurements*, realised by intake tubes leading, via ballast volumes, to the same gas analyser, which received air from these intakes sequentially in a switching cycle. By contrast, the tracer-release experiments all employed arrays of *line concentration sensors*, and, except in the study of McBain and Desjardins (2005), these sensors were all deployed at the same height. (Though Gao et al. (2009a) made measurements at six heights, they reported recovery rates obtained with the BLS method only for one.)

Third, WindTrax offers the user some choices how the wind and turbulence parameters are implemented. While the relationships between wind profile and wind statistics in the model are always forced to obey Monin–Obukhov similarity theory (MOST), there are different ways how these parameters can be entered. In McGinn et al. (2006, 2007), Flesch et al. (2007), and Harper et al. (2009), all required turbulence parameters are taken from a single sonic anemometer and WindTrax uses these to derive a vertical wind profile such that the roughness length, z_0, is fitted in each run for consistency between measured mean wind speed u, friction velocity u_* and Obukhov length L. As a consequence of inevitable measurement error as well as inhomogeneity and instationarity effects, z_0 can vary considerably and unrealistically from run to run. Where implausible values of z_0 occur, they are an indication that the shape of the wind profile near the ground may be inaccurate, which is a concern because all the emitted gas passes through the near-ground air layer on its way towards the measurement location, and consequently the speed of its horizontal advection may be in error. Alternatively, Laubach and Kelliher (2005a,b) and L08 first determine realistic z_0 values from the whole dataset of a five-point cup anemometer profile, corroborated by u_* and L from a sonic anemometer and visual assessment of site characteristics. These z_0 values do not vary at short time scale, only in response to observed variations of vegetation height (and where necessary, they vary with wind direction). This z_0 together with the wind speed from the uppermost cup anemometer and L from the sonic is entered into WindTrax. The model then computes u_* and ignores the value measured by the sonic anemometer, which is typically subject to 10% random error (and sometimes biased due to imperfect mounting). The different ways of entering the flow field parameters will lead to some differences in the calculated emission rates.

The previously quoted studies all assess the quality of the BLS results by comparing the modelled emission rate to either a known release rate or an alternative estimate of the unknown emission rate obtained by a different method. Here, a second aspect is added to the model assessment. We will investigate the field of the ratio of concentration, C (minus background, C_b) to emission rate, Q_c, in the alongwind-vertical plane (x,z) that contains the measurement mast $(y=0)$:

$$B(x,z) = \frac{C(x,z) - C_b}{Q_c} \qquad (1)$$

The normalised concentration field $B(x,z)$ is unambiguously constrained in WindTrax for a given source geometry and a given set of turbulence parameters. The specific experimental geometry of L08 provides the – so far unique – opportunity to compare the simulated B for five values of z and four values of x with reference B values obtained from measured concentrations and independently derived emission rates. Hence, the data of L08 will be re-analysed in the following to suit this purpose.

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